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Environmental trade-offs of relay-cropping winter cover crops with soybean in a maize-soybean cropping system

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Abstract

Winter camelina [*Camelina sativa* (L.) Crantz] and field pennycress [*Thlaspi arvense* L.] are oilseed feedstocks that can be employed as winter-hardy cover crops in the current cropping systems in the U.S. upper Midwest. In addition to provide multiple ecosystem services, they can be a further source of income for the farmer. However, using these cover crops is a new agricultural practice that has only been studied recently. The objective of this study was to assess and compare the environmental performance of a maize [*Zea mays* L.]-soybean [*Glycine* max (L.) Merr.] cropping system with different winter cover crops - camelina, pennycress, and rye (*Secale cereale* L.) - in the U.S. upper Midwest. Field experiments were carried out from 2016 to 2017 (2-year maize-soybean sequence) at three locations: Morris (Minnesota), Ames (Iowa), and Prosper (North Dakota). The environmental impact assessment was carried out using a "cradle-to-gate" life cycle assessment methodology. Four impact categories were assessed: global warming potential (GWP), eutrophication, soil erosion, and soil organic carbon (SOC) variation. Two functional units (FU) were selected: (1) 1 ha year−1, and (2) $1 net margin. When expressed with the FU ha yr−1, across the three locations cover crops had (a) lower eutrophication potential and water soil erosion, and (b) lower GWP if the cover crop was not fertilized with nitrogen. Camelina and pennycress were more effective than rye in reducing soil losses, while the three cover crops provided similar results for eutrophication potential. The results for the SOC variation were mixed, but the sequence with rye had the best performance at all locations. When expressed with the FU $ net margin, sequences including camelina and pennycress were overall the worst sequences in mitigating greenhouse gas emissions and nutrient and soil losses. This negative performance was mainly due to the seed yield reduction in the second year of the sequence for both the main cash crop (soybean) and the relayed-cover crop compared with the conventional sequence maize-soybean. Such result led to a lower net margin per hectare in the sequences including camelina and pennycress when compared with the control. The results of this study suggest that the introduction of camelina and pennycress as winter-hardy cover crops has a strong potential for reducing the environmental impacts of the maize-soybean rotation. However, a field management optimization of these cover crops in a relay-cropping system is needed to make them a sustainable agricultural practice.

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ABSTRACT

Winter camelina [*Camelina sativa* (L.) Crantz] and field pennycress [*Thlaspi arvense* L.] are oilseed feedstocks that can be employed as winter-hardy cover crops in the current cropping systems in the U.S. upper Midwest. In addition to provide multiple ecosystem services, they can be a further source of income for the farmer. However, using these cover crops is a new agricultural practice that has only been studied recently. The objective of this study was to assess and compare the environmental performance of a maize [*Zea mays* L.]-soybean [*Glycine* max (L.) Merr.] cropping system with different winter cover crops - camelina, pennycress, and rye (*Secale cereale* L.) - in the U.S. upper Midwest. Field experiments were carried out from 2016 to 2017 (2-year maize-soybean sequence) at three locations: Morris (Minnesota), Ames (Iowa), and Prosper (North Dakota). The environmental impact assessment was carried out using a "cradle-to-gate" life cycle assessment methodology. Four impact categories were assessed: global warming potential (GWP), eutrophication, soil erosion, and soil organic carbon (SOC) variation. Two functional units (FU) were selected: (1) 1 ha year^{-1}, and (2) $1 net margin. When expressed with the FU ha yr^{-1}, across the three locations cover crops had (a) lower eutrophication potential and water soil erosion, and (b) lower GWP if the cover crop was not fertilized with nitrogen. Camelina and pennycress were more effective than rye in reducing soil losses, while the three cover crops provided similar results for eutrophication potential. The results for the SOC variation were mixed, but the sequence with rye had the best performance at all locations. When expressed with the FU $ net margin, sequences including camelina and pennycress were overall the worst sequences in mitigating greenhouse gas emissions and nutrient and soil losses. This negative performance was mainly due to the seed yield reduction in the second year of the sequence for both the main cash crop (soybean) and the relayed-cover crop compared with the conventional sequence maize-soybean. Such result led to a lower net margin per hectare in the sequences including camelina and pennycress when compared with the control. The results of this study suggest that the introduction of camelina and pennycress as winter-hardy cover crops has a strong potential for reducing the environmental impacts of the maize-soybean rotation. However, a field management optimization of these cover crops in a relay-cropping system is needed to make them a sustainable agricultural practice.

1. Introduction

The introduction of cover crops into conventional crop rotations has shown to improve the overall sustainability of the agricultural production by providing multiple ecosystem services (Jordan and Warner 2010; Schipanski et al. 2014), both in the short term (e.g., moisture management, nutrient leaching reduction) and in the long term (e.g., erosion control, soil organic carbon preservation). Despite the extensively reviewed and recognized advantages of integrating cover crops in conventional cropping systems (Blanco-Canqui et al. 2015; Snapp et al. 2005), many U.S. farmers are still reluctant to adopt them. According to the 2017 U.S. Census of Agriculture, only 4.8% of the U.S. cropland had cover crops, although this was an increase of 1.5% from the previous census in 2012 (USDA 2019). Multiple factors are responsible for this

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situation, but direct and indirect costs, yield losses, and lack of economic incentives are often the main barriers (Bergtold et al. 2019; Dunn et al. 2016; Roesch-Mcnally et al. 2018; Singer et al. 2007; Snapp et al. 2005).

In colder agricultural regions such as the U.S. upper Midwest, the availability of winter-hardy cover crop species is limited, which also drastically limits double-cropping practices, especially in maize-soybean cropping systems (Snapp et al. 2005). Although double-cropping can still be a viable option for some short-season grain crops (Gesch et al. 2014; Johnson et al. 2017), relay-cropping or intercropping can be a more suitable alternative in regions with short growing season (Berti et al. 2015, 2017a). Relay-cropping is a form of temporal and, to some extent, spatial intensification in agriculture, where a crop is planted into another crop already established and their life cycles overlap for a period of time (Heaton et al. 2013). However, relay-cropping practices are challenging, as they create competition for resources between the main crop and cover crop (Ott et al. 2019). Also, there is often a limited time-window in fall for the establishment of cover crops in areas with a short growing season (Berti et al. 2015; Johnson et al. 2017). If the cover crop is used to provide multiple ecosystem services such as erosion control, nutrient management, and soil quality improvement, it is essential to ensure the soil is covered for the longest time possible. Winter-hardy cover crops can provide such function. To date, the winter-hardy cover crop most widely used in maize and maize-soybean cropping system in the U.S. upper Midwest is winter rye (Appelgate et al. 2017). Nevertheless, winter rye can reduce maize yield if not terminated two to three weeks before maize planting (Munawar et al. 1990). Hence, the need for other winter hardy-cover crops before maize in the upper Midwest region.

Introducing winter-hardy cash cover crops (winter cover crops for which biomass or grain are marketable products) would potentially increase the gross margin for the farmer while improving the environmental impact of agricultural practices in the U.S. upper Midwest region. Winter camelina and field pennycress have the agronomic characteristics to become an alternative to winter rye as cover crop in the U.S. upper Midwest. They can provide an additional source of income (cash cover crops) for farmers, in addition to enhancing plant diversification in the maize-soybean dominated cropping systems and provide nectars for pollinators (Berti et al. 2016; Obour 2015; Ott et al. 2019). Nevertheless, there are still significant barriers to the adoption of camelina and pennycress as cover crops (Cubins et al. 2019; Sindelar et al. 2017). First, more research is needed to identify optimal agronomic management practices for using camelina and pennycress as cover crops to minimize farmer's economic risk. Additionally, although their use in the biofuel and food industry could potentially be a viable option, the market for these oilseed species is still limited (Berti et al. 2016; Fan et al. 2013; Krohn and Fripp 2012; Moser 2012; Obour 2015; Sindelar et al. 2017). Highlighting broader potential environmental and ecosystem benefits provided through integrating these winter cover crops in a relay-cropping system might foster their possible adoption by farmers for large-scale cultivation.

The assessment of carbon footprint and soil quality in intercropped systems is a growing field of research in agricultural studies, which attempts to quantify the benefits of intercropping and implement more efficient cropping systems (Cong et al. 2015; Hauggaard-Nielsen et al. 2016; Wang et al. 2020). However, holistic environmental impact assessment studies of intercropping systems are limited, mainly due to the challenges and complexity of modelling the environmental impacts of multiple crops growing simultaneously in the same field. While the agronomic effects of cover crops in double or relay-cropped systems on nutrient management and soil fertility have been extensively examined in the literature, life cycle assessment (LCA) studies investigating the overall environmental impact of cover crops in cropping systems are still limited. In the European context, Iogos et al. (2016) used the LCA methodology to study an innovative double-cropping system maize-rye, where the winter cover crop was used for bioenergy purposes. In another study, Naudin et al. (2014) analyzed a cereal-legume [pea (*Pisum sativum* L.)-wheat (*Triticum aestivum* L.)] intercropping system, although without a cover crop. In addition, a LCA of double-cropping systems with legume, non-legume, and mixture cover crops was conducted in Switzerland (Prechsl et al. 2017). In the U.S., the impact of cover crops on maize and maize-soybean rotations within an LCA for biofuel production was discussed in Kim and Dale (2005) and Kim et al. (2009). However, to date, only one study investigated the LCA of relay-cropping camelina with soybean in the U.S. upper Midwest (Berti et al., 2017a). A number of agricultural LCA studies on spring camelina (Krohn and Fripp 2012; Miller and Kumar 2013) and pennycress (Fan et al. 2013) have been carried out previously to assess their environmental impact when grown as main full-season crops for biofuel production.

The objective of this study was to assess and compare the environmental performance of a 2-year maize-soybean rotation without winter crops (winter fallow) and with camelina, pennycress or rye as winter crops. The study provides a quantitative assessment of the environmental trade-offs of cropping sequences with or without cover crops, contributing to evaluate the overall sustainability of ecological intensification practices - such as relay-cropping and winter cover crops - in conventional cropping systems. The findings of this analysis also provide useful information to local and regional decision-makers (e.g., farmers, consultants, and policymakers) to assess benefits and obstacles of introducing new winter-hardy cover crops in the current U.S. upper Midwest agricultural landscape.

2. Methodology

2.1. Cropping systems and experimental design

In the field experiments conducted by Mohammed et al., 2020a, multiple winter-hardy cover crops were introduced within a 2-year maize-soybean cropping system. Camelina, pennycress, and rye were interseeded into standing maize at R4, R5, and R6 development stages for maize. The following season soybean was relay-seeded into standing camelina and pennycress, while rye was terminated with glyphosate two weeks before sowing soybean. A control scenario with the conventional maize-winter fallow-soybean sequence was also included in the experiment. A description of the cropping sequences employed in the field experiments in all three locations is shown in Fig. 1. The field experiments were carried out from 2016 to 2017 in three locations (rain-fed environment): Morris (Minnesota), Ames (Iowa), and Prosper (North Dakota).

The experimental design at Ames and Morris was a randomized complete block design (RCBD) with a split-plot arrangement and four replicates, and at Prosper, a RCBD with four replicates. An average of the data from the four replicates was used in the environmental assessment. Maize and soybean were seeded with 76-cm row spacing in all locations, using cultivars or hybrids adapted to the conditions of the respective geographical area.

Crop management practices were different in each location. In the Morris and Ames experiments, a similar fertilization protocol (conventional fertilization rate) was adopted, while at Prosper a low-input management over the 2-year sequence was chosen. In addition, maize residue management was different in the three locations: in Morris and Prosper, 70% and 95% of the aboveground maize residue were removed in Year 1, respectively, to facilitate the cover crop establishment, while in Ames maize residue was not removed from the field. Main characteristics of the three experimental sites and field management protocols adopted are summarized in Table 1. Further details on the experimental design and agronomic management practices used can be found in Mohammed et al., 2020a.

2.2. Life cycle assessment

The environmental impact assessment was carried out using a LCA methodology following the ISO 14040 standard (ISO 2006). The

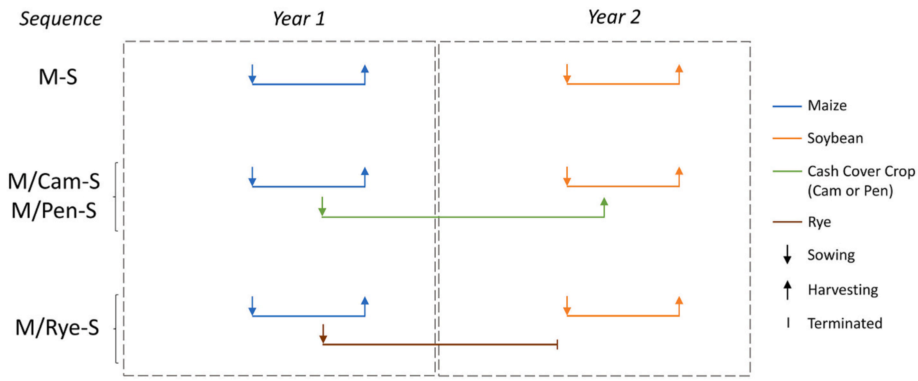


Fig. 1. Cropping sequences assessed in the environmental impact assessment.

environmental impact of the sequences with winter cover crops in the maize-soybean system was assessed and compared with a conventional crop sequence for each of the three locations considered in the study.

2.2.1. Scope of the study and functional unit

The system boundary was set from cradle to farm gate, which means that all farm inputs (e.g., production of seeds, fertilizers, pesticides), field operations, crop outputs (grain yield and biomass) and farm direct emissions (e.g., N-related emission, phosphates, carbon dioxide, pesticides) were included in the assessment (Fig. 2).

To quantify and compare the environmental performance of the different crop sequences considered in this study, two functional units were selected: (1) 1 ha year^{-1}, and (2) $1 (USD) net margin. The former unit relies on the system's inputs and does not take into account the crop yield (except for the estimate of machinery use and crop residue), which is the variable that is most affected by local environmental conditions (e.g., soil moisture, weather conditions) in rain-fed agricultural systems. Such land management function provides an overview of the spatial and temporal aspects of environmental impacts related to the agricultural practice (Nemecek et al. 2011). In addition, as stated in Berti et al., 2017a, most farmers identify type and amount of inputs before the beginning of the season as per the potential seed yield they foresee to obtain. Therefore, an area-based unit is particularly useful to inform farmers on the decision-making process related to crop planning. The latter unit, $1 net margin, associates the environmental impacts with the farmer's net margin. It directly depends on yield revenues and production costs. Land rent equivalent was not included in the total costs to avoid potential negative net margins that could make the comparison of the sequences more complicated to interpret. In comparison with the land-based unit, this functional unit is strongly affected by the environment characteristics and conditions and is more time-dependent. Nevertheless, through including an economic component in the assessment process, this functional unit provides an additional element to compare the overall sustainability of different cropping sequences, as well as offers a further key element for the farmer's decision-making process (Notarnicola et al. 2017). Employing multiple functional units provides a better understanding of the overall assessment process (Nemecek et al. 2011). However, the multifunctionality of the agricultural systems makes the choice of a functional unit a challenging task, which strongly affects the results of the assessment (Caffrey and Veal 2013), particularly when intercropping systems are assessed (Goglio et al. 2018; Naudin et al. 2014).

2.2.2. Life cycle inventory

The LCA model in this study was developed using SimaPro 9.0.0.35 software (PRé Consultants, 2019). The life cycle inventory (LCI) was built by using multiple data sources, such as field experiment data and Ecoinvent v3 database (Wernet et al. 2016) (primary sources for inputs), models (primary source for outputs), literature review, and expert opinions. While databases were mostly used to model the production phase of farming inputs before their use in the field, direct field emissions were estimated using multiple models, including air emissions (nitrous oxide, nitric oxide, ammonia, carbon dioxide, methane, and pesticides), superficial and ground water emissions (nitrates, phosphates, and pesticides), and soil emissions (pesticides).

Nitrous oxide (N_2O) emissions from nitrogen-fertilization in agricultural soils can be estimated by using emission factors or more accurate empirical or processed-based models (Goglio et al. 2018). To date, it is still a common practice to use emission factors in LCA studies due to the complexity and resources required to build and run more advanced models (Peter et al. 2016). Nitrous oxide emission models can be divided into two macro-groups, according to the function they use to estimate the emissions from the N-input: linear or exponential. Even though linear models are still used to a great extent in particular for large scale estimations (e.g., IPCC 2006), several authors have reported that the relationship between N inputs and N_2O emissions might not be linear but exponential (Grace et al. 2011; Kim et al. 2013; Philibert et al. 2012; Shcherbak et al. 2014). In this study, we employed an exponential model to estimate N_2O and NO (nitric oxide) emissions proposed by Bouwman et al. (2002a). This methodology was developed by identifying and then modelling the most significant factors affecting N_2O and NO emissions from a large dataset of field measurements found in the literature (Bouwman et al. 2002b). According to Bouwman et al. (2002a), annual N_2O emissions are significantly affected by rate and type of N-fertilizer, crop type, soil texture, soil organic carbon, soil drainage, soil pH, climate type, and length of the experiment, while NO emissions mainly depend on rate and type of N-fertilizer, soil organic carbon, and soil drainage. Several studies (Philibert et al. 2012; Kim et al. 2013; Shcherbak et al. 2014; Peter et al. 2016;) confirm the overall quality of the estimation generated by this model. A similar model developed by the same authors (Bouwman et al. 2002c) was used to estimate the ammonia (NH_3) losses from mineral N-fertilization. In this case, the significant factors identified by the authors were fertilizer type, rate and application mode, type of crop, soil pH, soil cationic exchange capacity (CEC), and type of climate. A reduction factor of −53% was applied to the ammonia emissions when urea with urease inhibitor was used in the experiments (Cantarella et al. 2018).

Greenhouse gases emissions (i.e., carbon dioxide, methane, and nitrous oxide) from fossil fuels combustion due to machinery field operations were estimated according to the US-EPA (2018) emission factor guidelines for greenhouse gases (GHGs) emissions inventories. The amount of fossil fuels used for field operations were calculated according to field experiment data, literature review, and expert opinion. Carbon dioxide (CO_2) emissions from urea application were calculated by applying the Tier-1 IPCC (2006) emission factor.

Pesticides emissions to soil were estimated according to the Product Environmental Footprint Rules Guidance (European Commission 2018): 90% to soil, 9% to air and 1% to water, which are assumed to be

Table 1
Site characteristics, field management and inputs for the three locations analyzed in this study.

Location (city, state)	Ames, Iowa	Morris, Minnesota	Prosper, North Dakota
Site Characteristics			
Geographical location	42.00, −93.73	45.67, − 95.80	46.97, −97.05
Annual total precipitation (mm)	956 *(2016)*; 755 *(2017)*	682 *(2016)*; 685 *(2017)*	386* *(2016)*; 384* *(2017)*
Annual average temperature (°C)	11.5 *(2016)*; 11.1 *(2017)*	7.8 *(2016)*; 6.9 *(2017)*	7.2 *(2016)*; 5.7 *(2017)*
Soil texture	Loam	Loam	Silt loam
Soil organic carbon (% SOC)	1.8	3.4	2.4
Drainage	Good	Good	Poor
Field Management			
Management type	Conventional input, minimal residue removal	Conventional input, medium residue removal	Low input, high residue removal
Tillage	Conventional tillage in Year 1: disk plow and spring field cultivation. No-till in Year 2	Conventional tillage in Year 1: disk and chisel plow and spring field cultivation. No-till in Year 2	Reduced tillage in Year 1: chisel plow and spring field cultivation. No-till in Year 2
Residue management	No maize, soybean and rye residue removal. 100% camelina and pennycress residue removal	70% maize residue removed. No soybean and cover crop residue removal	95% of maize residue removed. No soybean and cover crop residue removal
Agricultural Inputs			
Fertilization rate: N-P-K (kg ha^{-1})	Maize: 168–123–112	Maize: 150–70–30	Maize: 124–0–0
	Camelina and pennycress: 78–34–34	Camelina and pennycress: 78–34–34	Camelina and pennycress: 0–0–0
	Soybean: 0–0–0	Soybean: 0–0–0	Soybean: 0–0–0
Fertilizer's type	Diammonium phosphate, urea, sulfur-coated urea, muriate of potash	Diammonium phosphate, urea, ammonium sulfate, muriate of potash, zinc sulfate	N-(n-butyl) thiophosphoric triamide (NBPT)-treated urea
Herbicide/ pesticide (kg a.i. ha^{-1})	Maize: glyphosate (2.24), pendimenthalin (1.38)	Maize: glyphosate (1.1)	Maize: glyphosate (1.1)
	Soybean: lactofen (0.56)	Soybean: glyphosate (1.1), lambdacyhalothrin (0.01)	Soybean: N/A
	Rye (*to terminate*): glyphosate (2.24)	Rye (*to terminate*): glyphosate (2.13)	Rye (*to terminate*): glyphosate (2.24)
Seeding rate, pure live seed (kg ha^{-1})	Maize: 30.4	Maize: 30.4	Maize: 30.0
	Camelina: 11.2 Pennycress: 16.8 Rye: 84.1 Soybean: 77.0	Camelina: 11.2 Pennycress: 16.8 Rye: 84.1 Soybean: 77.0	Camelina: 11.2 Pennycress: 16.8 Rye: 84.1 Soybean: 79.6
Machinery			
Machinery use: (a) = fuel (kg ha^{-1}); (b) = electricity (kWh ha^{-1}); (c) = natural gas (m^3 ha^{-1})			
M-S	(a) 69.7; (b) 40.7; (c) 108.8	(a) 76.9; (b) 38.3; (c) 102.5	(a) 68.9; (b) 32.5; (c) 86.8
M/Cam-S	(a) 95.7; (b) 43.8; (c) 109.3	(a) 100.7; (b) 41.4; (c) 100.8	(a) 92.4; (b) 31.6; (c) 81.8
M/Pen-S	(a) 95.8; (b) 41.2; (b) 100.4	(a) 101.0; (b) 44.9; (c) 100.1	(a) 92.8; (b) 37.8; (c) 86.2
M/Rye-S	(a) 72.6; (b) 40.1; (c) 107.1	(a) 78.9; (b) 39.1; (c) 104.6	(a) 70.9; (b) 32.2; (c) 86.1

* Missing data: monthly precipitation in Jan-Mar and Nov-Dec 2016, and Jan-Mar and Nov-Dec 2017.

reasonable temporal estimations "based on expert judgment due to current limitations" (European Commission 2018:72).

The eutrophication process mainly depends on the availability of two nutrients in water, nitrogen and phosphorus (Conley et al. 2009; Dodds and Smith 2016). Annual nitrate leaching was estimated through the SQCB-NO_3 model (Emmenegger et al. 2009; Nemecek and Schnetzer 2011), which was recommended in agricultural LCA for non-European countries (Nemecek et al. 2016; Nemecek and Schnetzer 2011). This model considers weather conditions, soil properties (clay content, bulk density, carbon/nitrogen ratio, and organic nitrogen content), plant characteristics (rooting depth and nitrogen uptake), and nitrogen fertilization rates. Phosphate and phosphorus emissions to surface and ground water were calculated based on the Salca-P Model (Nemecek and Schnetzer 2011). Salca-P model estimates three types of emissions to water: (1) soluble phosphate leaching to ground water, (2) soluble phosphate run-off to surface water, and (3) water erosion of soil particles containing phosphorus to surface water.

2.2.3. Life cycle impact assessment method

The effect of cover crops on the 2-year maize-soybean rotation was evaluated in the life cycle impact assessment (LCIA) phase for two categories of impact: (1) 100-year global warming potential (GWP) according to IPCC 2013 (Myhre et al. 2013); (2) eutrophication impact according to the TRACI 2.1 methodology (Bare 2011). Two additional categories were also included in the assessment phase, (3) soil erosion, and (4) soil quality.

Soil erosion was added to the environmental assessment since one of the multiple benefits of using winter-hardy cover crops is their ability to prevent soil erosion in agricultural soils. Winter-hardy cover crops ensured a living cover during the fall and spring seasons and a protection due to residues over the winter period (Berti et al. 2017b; Snapp et al. 2005). Agricultural soils are affected by two types of erosion agents, water and wind, which are modelled separately. In this study, two USDA-developed models were employed to quantify the annual soil erosion: RUSLE2, for water erosion (Foster et al. 2003a, 2003b) and WEPS, for wind erosion (USDA 2016). However, only the results of the water erosion model are included in this paper. The WEPS model is not designed to model relay-cropping systems. Nevertheless, a simulated double-cropping system (cover crop sown after maize harvest in Year 1 and harvested or terminated before sowing soybean in Year 2) was run to assess the possibility of using the results as a very conservative estimate of wind soil erosion rates. However, such simulation produced inconsistent results for all three locations, which led to the final exclusion from this publication.

Cover crops are often introduced to increase the overall soil quality of cropping systems (Villamil et al. 2006). A number of scholars have indicated that soil organic carbon (SOC) changes can be used as a proxy soil quality indicator in LCA (Brandão et al. 2011; Goglio et al. 2015; Milà i Canals et al. 2007). Preserving SOC is a critical factor for agricultural production, particularly in North America, where 20% to 75% of the original top soil carbon (0–30 cm) was lost due to the conversion of native prairie to agriculture (Lajtha et al., 2018). In this study, the impact of the cropping sequence on the SOC stock was assessed according to the methodology of the *minimum residue return rate* developed

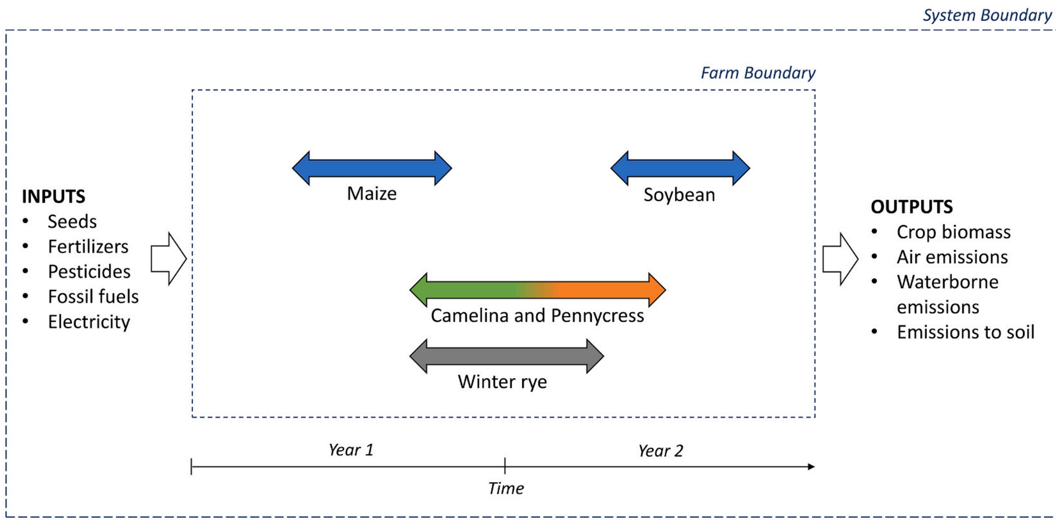


Fig. 2. Material and temporal system boundaries for the studied sequences. All sequences have the maize-soybean 2-year rotation in common, without winter cover crop (control) or with winter camelina, field pennycress or winter rye as winter cover crop.

by Johnson et al. (2006, 2009, 2013, 2014). The minimum residue return rate is the minimum amount of aboveground residues needed to maintain the baseline levels of C in agricultural soils. Building on their work on the SOC variation in US agricultural soils, Johnson et al. (2009) identified an overall minimum source C (MSC) value of 2.5 ± 1.7 Mg C ha^{-1} year^{-1} from aboveground residues to maintain initial SOC levels. Therefore, in this paper, the total amount of C provided to soil by aboveground residues for each crop sequence was estimated by calculating the C input from crop residues and compared with the reference value provided by Johnson et al. (2009).

2.2.4. *Uncertainty assessment*

The robustness of the LCA results was tested through uncertainty assessment by using a Monte Carlo Analysis (10,000 runs, 95% confidence interval). Unless variations in field data or modelled estimates were specifically reported, inputs and farm-level emissions were assumed to have a lognormal distribution and the standard deviation was determined by using Pedigree matrix (Weidema and Wesnæs 1996) or basic uncertainty values in SimaPro (Goedkoop et al. 2016). When an error range was provided for outputs estimated through models (e.g. N_2O field emission model), a triangular distribution was selected.

The uncertainty for the results of modelling water erosion was determined according to the model's accuracy values presented by USDA (2001). Finally, the results of the SOC variation were reported showing the potential variation in the annual carbon input necessary to maintain SOC levels (± 1.7 Mg C) according to Johnson et al. (2009).

3. Results

3.1. *Global warming potential (GWP)*

The average GWP 100-year calculated in the LCA was 2470 kg CO_2

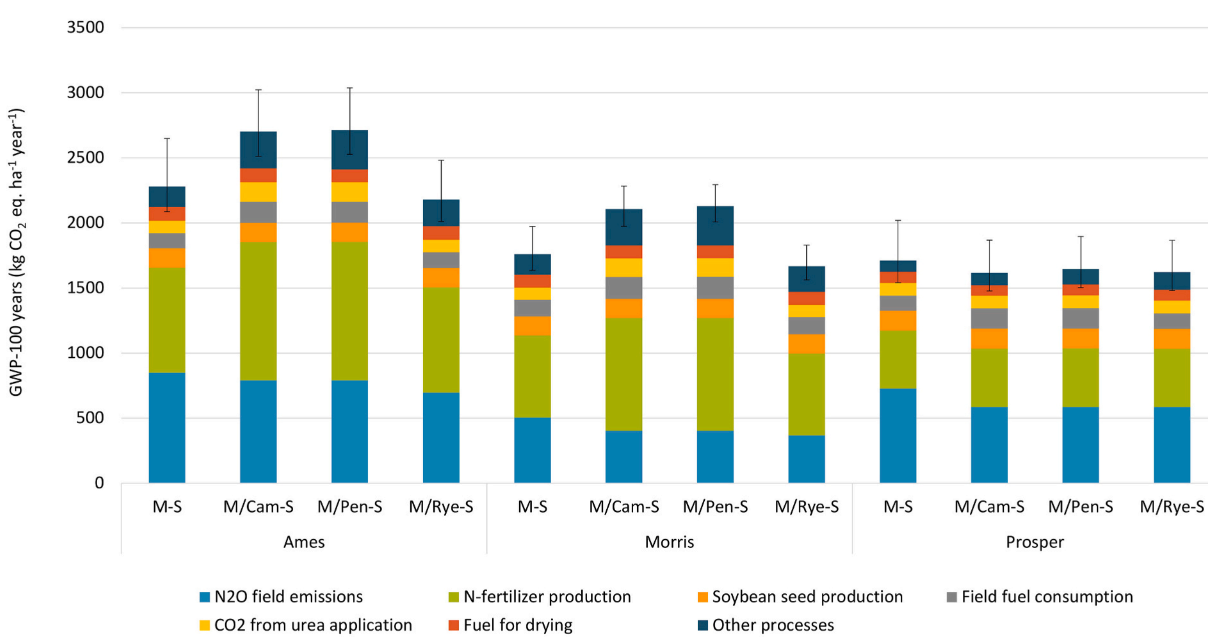


Fig. 3. Global warming potential (GWP-100-year) and relative contribution of process inputs and outputs for four sequences in three locations. The results are expressed according to the land-based functional unit: ha year^{-1}. Maize-soybean = M-S, maize/camelina-soybean = M/Cam-S, maize/pennycress-soybean = M/Pen-S, and maize/rye-soybean = M/Rye-S.

eq. ha^{-1} $year^{-1}$ in Ames, 1920 kg CO_2 eq. ha^{-1} $year^{-1}$ in Morris, and 1650 kg CO_2 eq. ha^{-1} $year^{-1}$ in Prosper (Fig. 3a). These values are consistent with a well-known association between agricultural GHG emissions and fertilization practices: The highest GWP was found in the site with the highest N-fertilization rates (Ames, 246 kg N ha^{-1} over a 2-year period), while Prosper (where the N applied over a 2-year period was 124 kg N ha^{-1}) had the lowest GWP. In all sequences and at all three locations, the N-fertilization (fertilizer production and field emissions due to N-fertilizer application) contributed to approximately 70% of the total annual GWP-100 years, a value in line with previous findings (Kim and Dale 2008; Wightman et al. 2015). The main source of uncertainty for the GWP impact was associated with the estimate of N_2O field emissions, which had an uncertainty range between −40% and + 70% (Bouwman et al. 2002a). As shown in Fig. 3, Morris location showed the lowest uncertainty among the three sites, which was mainly caused by a lower N_2O field emission for all sequences in this location compared with that of Ames and Prosper.

Nitrous oxide field emissions due to N-fertilizer application were the main source of GHG emissions in Prosper (average of all treatments: 37.6% of the GWP-100 years) and all sequences with cover crops had lower N_2O field emissions than the control (average reduction of N_2O field emissions in treatments with cover crop = −19%). For Ames and Morris, which had higher N fertilizer rates than those at Prosper, the overall GHGs emissions of N-fertilizers production phase (urea and diammonium phosphate) contributed the most to the GWP results (with the only exception of the control in Ames). Nitrogen fertilizer production requires energy to convert a N_2 molecule to NH_3 and the majority of nitrogen fertilizer plants use fossil fuels (natural gas) as energy source to produce urea (Gellings and Parmenter 2004). At all locations and in all sequences, CO_2 emissions related to urea application had a lower contribution to the total GWP than N_2O field emissions and N-fertilizer production (between 4.2% and 6.8% of the total GWP).

Among the elements not related to the nitrogen fertilization that contribute to the GWP, machinery fuel production and consumption and soybean seed production had the highest GHG emissions (values between 5 and 10% of the total GWP), followed by the emissions related to fuel (natural gas and propane) used to dry maize grain (3–6%).

Overall, the LCA shows that when the cover crops did not receive N-fertilization, the GWP of sequences with cover crops was lower compared with the control. This trend was consistent in Prosper for all sequences tested and only for the M/Rye-S sequence in Ames and Morris (Fig. 3). Conversely, when the cover crops were fertilized with N, the control had lower GWP than the other treatments, due to a higher contribution of N-fertilizer production to the GWP of the sequences with cover crops.

When introducing the economic component into the environmental assessment (economic functional unit), the LCA results changed substantially. The M/Cam-S and M/Pen-S sequences showed inferior performance (higher GWP $\$^{-1}$) when the LCA output was expressed in the net margin functional unit (Fig. 4) compared with a land-based functional unit. Even in Prosper, where all sequences with cover crops had lower GWP ha^{-1} than the maize-soybean sequence in the land-based functional unit, the GWP $\$^{-1}$ for the M/Cam-S and M/Pen-S sequences were approximately doubled compared with the control (Fig. 4). In addition, in all three locations the M/Rye-S sequence did not have the least GWP $\$^{-1}$ (mainly due to the added cost of seeding and terminating the cover crop), although values were very close to the maize-soybean GWP (which had the lowest emissions per $).

3.2. Eutrophication

The maize-soybean sequence including cover crops had lower eutrophication potential than the control scenario in all three locations. In Prosper, where the cover crops were not fertilized, camelina and pennycress were the most effective cover crops to reduce NO_3-N loading. Pennycress had higher biomass yield than camelina, which resulted in higher NO_3-N uptake and lower NO_3-N released to surface and ground waters. In Ames and Morris, rye was the best cover crop to mitigate eutrophication. However, despite the N-P-K fertilization of camelina and pennycress in these locations, these two cover crops still had similar results as rye, which means that the additional N and P fertilization was mostly taken up by the cover crops. These two factors – nitrate leaching and phosphorus emissions through water erosion – were responsible for about 90% of the eutrophication potential (88.4–93.5%). The uncertainty associated with the nitrates and phosphorus loss estimates contributed the most to the overall uncertainty for the eutrophication impact category, making it difficult to differentiate the results between the sequences, in particular those with cover crops.

The erosion of particulate phosphorus contributed the most to the overall eutrophication potential for the control in Ames (52.8%) and Morris (48.6%), where maize received a P fertilization of 123 kg ha^{-1} and 70 kg ha^{-1}, respectively (Table 2). Camelina and pennycress were fertilized with 34 kg P_2O_5 ha^{-1} in Year 2, but the reduction of soil erosion balanced out the potential release to surface waters of this extra load of phosphorus. In fact, M/Cam-S and M/Pen-S sequences had the best management of phosphorus losses in both Ames and Morris locations (Table 2).

Conversely, the assessment results suggest that the 78 kg ha^{-1} of N-fertilizer provided to camelina and pennycress in Year 2 were partially used by these cover crops. The M/Cam-S and M/Pen-S sequences had slightly greater nitrate leaching than the control in both Ames and Morris locations (Table 2).

In Prosper, phosphorus was not applied because soil P levels were above the critical level of 16 mg kg^{-1} recommended for fertilization with P in North Dakota. All treatments received the same amount of nitrogen (124 kg ha^{-1}) in Year 1. The very low levels of water erosion in this site, in addition to no external P inputs, considerably reduced the contribution of P releases to the total eutrophication potential to a

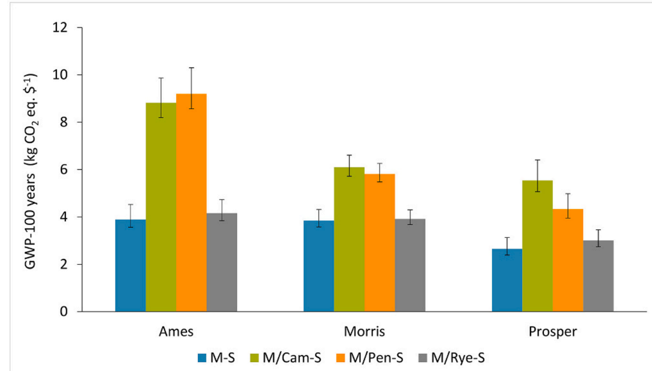


Fig. 4. Global warming potential (GWP-100-year) for four sequences in three locations. The results are expressed according to the economic functional unit: $ net margin. Maize-soybean = M-S, maize/camelina-soybean = M/Cam-S, maize/pennycress-soybean = M/Pen-S, and maize/rye-soybean = M/Rye-S.

Table 2
Contribution of phosphorus erosion and nitrate leaching to the eutrophication potential of four crop sequences in three locations. Maize-soybean = M-S, maize/camelina-soybean = M/Cam-S, maize/pennycress-soybean = M/Pen-S, and maize/rye-soybean = M/rye-S.

Sequence	Ames	Morris	Prosper	Ames	Morris	Prosper
	Phosphorus erosion			Nitrate leaching		
	kg N eq. ha^{-1} $year^{-1}$			kg N eq. ha^{-1} $year^{-1}$		
M-S	49.5	38.5	4.5	37.1	35.6	32.5
M/Cam-S	30.8	24.2	2.7	41.3	36.5	31.5
M/Pen-S	28.3	21.8	2.3	39.9	38.5	31.5
M/Rye-S	36.2	27.5	3.1	31.8	32.8	30.9

6.2–11.3%.

When the results of the LCA are expressed per $ net margin (Fig. 5b), in all locations, M/Cam-S and M/Pen-S sequences showed a similar response to what was previously described for the GWP category. This trend is particularly evident in Ames and Prosper, where the eutrophication potential per $ for the control is respectively 40% and 52% lower than the M/Cam-S sequence. In Morris, the gap between the control and the M/Cam-S and M/Pen-S sequences was within a 10% margin. The M/Rye-S sequence had the lowest eutrophication potential per $ in Ames and Morris. In Prosper, the control showed the lowest eutrophication potential per $, while M/Cam-S, M/Pen-S, and M/Rye-S sequences had 41%, 45%, and 8% higher eutrophication potential than the control, respectively.

3.3. Soil water erosion

In all three locations, the M/Pen-S sequence had the highest soil erosion reduction, followed by M/Cam-S and M/Rye-S sequences (Fig. 6a). The M/Pen-S sequence reduced the soil losses by almost half when compared with the control (50% in Prosper, 43% in Ames and 45% in Morris), while the sequences M/Cam-S and M/Rye-S reduced soil losses by 39% and 33% of the control, respectively, and averaged across all locations. Between the cover crops considered, the M/Rye-S sequence was the least effective in curbing erosion, likely because it covered the soil for a shorter period than camelina and pennycress. Rye was terminated in late April or early May of Year 2 before seeding soybean, while pennycress and camelina were harvested between late June and late July of Year 2. The M/Pen-S sequence reduced soil losses more than the M/Cam-S sequence due to higher biomass production in all three locations. Although all locations demonstrate a degree of uncertainty for the soil loss category, Prosper shows the highest uncertainty in soil erosion values. This is due to accuracy of the estimates provided by the RUSLE2 model, which have a 50% uncertainty for values between 1.1 and 9 Mg ha^{-1} (such as for Ames and Morris locations) and much higher (up to 500%) for soil losses lower than 1.1 Mg ha^{-1} (USDA 2001).

When the soil erosion values per hectare are divided by the net margin for the farmer (i.e., economic functional unit), results changed (Fig. 6b). Only in Morris the sequences with cover crops had lower soil losses per $ of net margin than that of the control. The soil loss estimate was 13.2 kg per $ of net margin generated for the control, while the

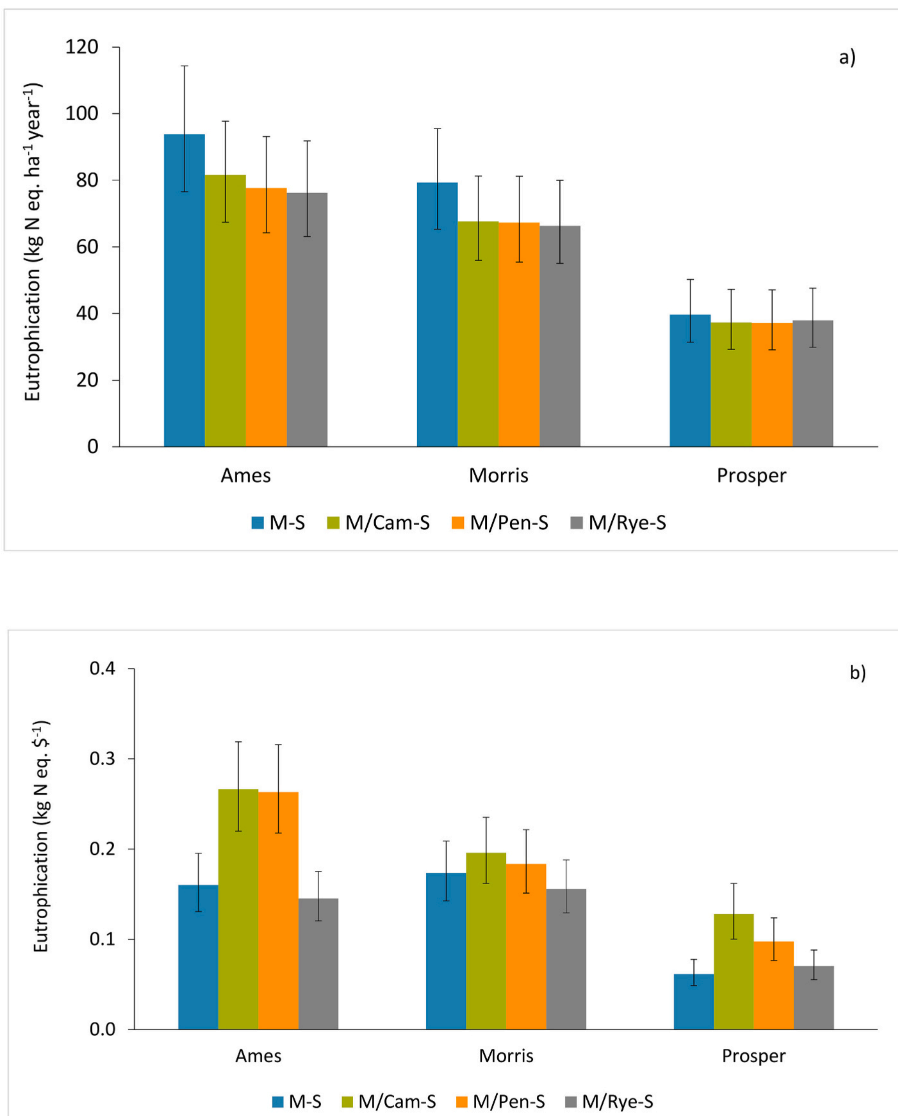
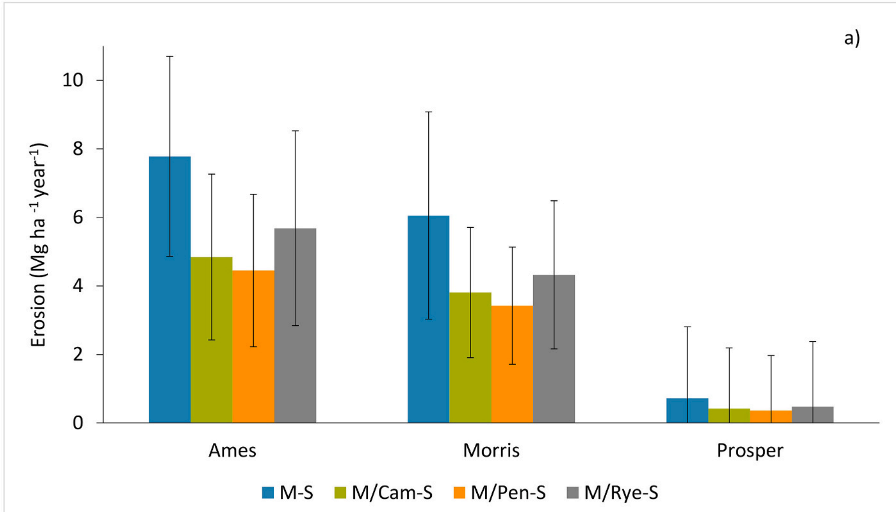


Fig. 5. Eutrophication assessment for four crop sequences in three locations. 5a) results based on the land-based functional unit ha year^{-1}, 5b) results based on the economic functional unit $ net margin. Maize-soybean = M-S, maize/camelina-soybean = M/Cam-S, maize/pennycress-soybean = M/Pen-S, and maize/rye-soybean = M/rye-S.



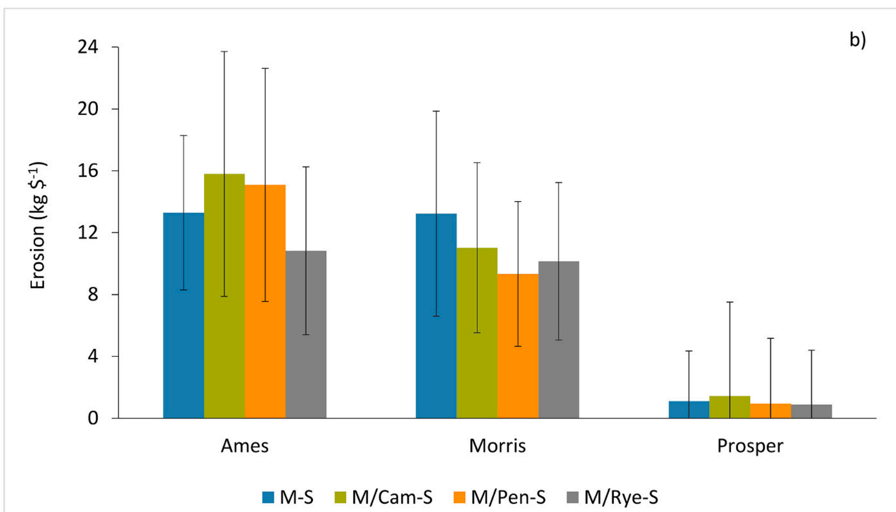


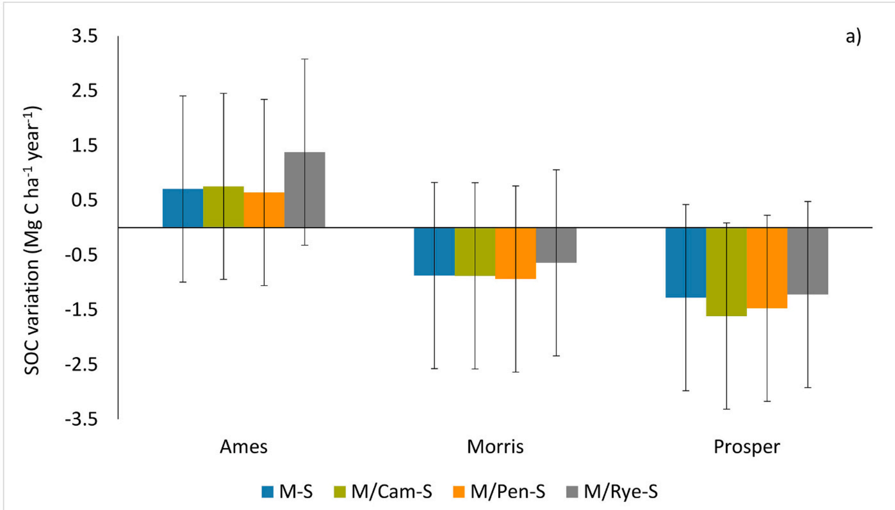
Fig. 6. Results of soil erosion impact assessment for four sequences in three locations. 5a) results based on the land-based functional unit ha year^{-1}, 6b) results based on the economic functional unit $ net margin. Maize-soybean = M-S, maize/camelina-soybean = M/Cam-S, maize/pennycress-soybean = M/Pen-S, and maize/rye-soybean = M/Rye-S.

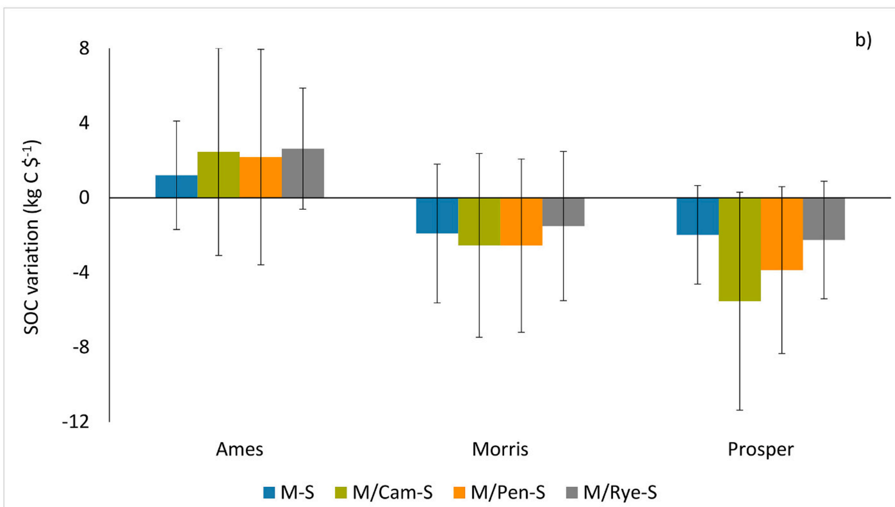
cover crops ranged between 9.3 kg \$$^{-1}$ and 11 kg \$$^{-1}$. In Morris, for both functional units, cover crops had better results than the control in limiting erosion, but when using the net margin unit, the gap between sequences with cover crops and the control was reduced. Conversely, in Ames, camelina and pennycress were not cost-effective at limiting soil losses (15.1 and 15.8 kg \$$^{-1}$, respectively) when compared with the control, which had an estimated soil loss of 13.3 kg \$$^{-1}$. The M/Cam-S sequence was also the least effective option in Prosper (14.4 kg \$$^{-1}$), even if the sequence with pennycress had lower soil losses than the control (9.5 versus 11.2 kg \$$^{-1}$). For both Ames and Prosper, rye was the most cost-effective cover crop (10.8 and 8.9 kg \$$^{-1}$, respectively) when expressing the assessment results per $ net margin.

3.4. Soil organic carbon (SOC) variation

The residue field management in Morris (70% maize residue removal in Year 1) and Prosper (95% maize residue removal in Year 1) negatively affected the SOC levels, with an average organic carbon loss of 0.83 Mg C ha^{-1} year^{-1} and 1.40 Mg C ha^{-1} year^{-1} in Morris and Prosper, respectively (Fig. 7a). The choice of removing a large part of the maize residue to facilitate the cover crop establishment led to an overall SOC depletion over the 2-year rotation. The contribution of the cover crop and soybean residues (which were not removed) in Year 2 was not able to offset the SOC debt created in Year 1 and meet the critical C return rate of 2.5 Mg C ha^{-1} per year proposed by Johnson et al. (2009). For both Morris and Prosper, all sequences had a SOC debt in Year 2 as well. For these two locations, M/Cam-S and M/Rye-S sequences SOC debt were lower than the control. This overall SOC debt can be ascribed to a limited C return rate from soybean and the cover crops in the sequence.

Similarly, at Ames SOC decreased for Year 2, with the only exception being the M/Rye-S sequence, in which cover crop biomass was not removed as in the other two sequences, M/Cam-S and M/Pen-S. This contributed to a SOC credit of 0.35 Mg C ha^{-1} in the M/Rye-S sequence. Not harvesting maize residue generated a carbon credit (2.1–2.5 Mg C ha^{-1}) in Year 1, that balanced out the debt in Year 2 for the sequences M/Cam-S, M/Pen-S, and the control. The average annual carbon credit in Ames ranged between 0.64 Mg C ha^{-1} for the M/Pen-S sequence and 1.38 Mg C ha^{-1} for the M/Rye-S sequence (Fig. 7a). The M/Rye-S sequence outperformed all other crop sequences analyzed due to a higher total biomass availability over the 2-year period. Even if camelina and pennycress residues were not removed, the M/Rye-S sequence would still have had higher SOC credit because rye biomass yield in Year





Figs. 7. Soil organic carbon (SOC) variation for four sequences in the three locations. 7a) results based on the land-based functional unit ha year^{-1}, 7b) results based on the economic functional unit $ net margin. Maize-soybean = M-S, maize/camelina-soybean = M/Cam-S, maize/pennycress-soybean = M/Pen-S, and maize/rye-soybean = M/Rye-S.

2 was 2–3- fold greater than pennycress and camelina biomass yield. Similar results were obtained in Morris and Prosper, where the M/Rye-S sequence limited SOC losses. The error bars provided in Fig. 7a show the variation (± 1.7 Mg C) from the threshold of 2.5 Mg C. This means that the cropping system studied could potentially have neutral or positive SOC variations in soils that require low C inputs, even in Morris and Prosper. On the contrary, in soils that require inputs higher that 2.5 Mg ha^{-1}, the biomass produced and the field management adopted in Ames could not be sufficient to maintain SOC levels in soil.

The LCA results on SOC changed substantially when expressed per $ net margin (Fig. 6b) compared with the land-based functional unit. For Ames, all sequences with cover crops outperformed the control by doubling the annual SOC credit generated per dollar; 1.2 kg C $\$^{-1}$ for the control and 2.2–2.6 kg C $\$^{-1}$ for sequences with cover crops. The differences in SOC change between the sequences with cover crops were smaller than using the functional unit of kg C ha^{-1}. In Morris and Prosper, which had a net SOC depletion in all sequences, the M/Cam-S and M/Pen-S sequences had much greater SOC losses when the results were expressed in the economic functional unit rather the area-based unit. This trend is particularly clear in Prosper, where the control had the least SOC losses per $ net margin. The M/Rye-S sequence was similar to the control, 2.3 kg C $\$^{-1}$ and 2.0 kg C $\$^{-1}$, respectively, while the annual SOC loss was two and almost three times greater than the control, in the M/Pen-S (3.9 kg C $\$^{-1}$) and M/Cam-S (5.5 kg C $\$^{-1}$) sequences, respectively. The M/Rye-S was still the best sequence to mitigate SOC losses, but only in Ames and Morris. Overall, no clear trend on SOC variation can be identified when the LCA results are expressed in the economic functional unit, which - as mentioned before - is more dependent on the annual variability of the local environmental conditions (precipitation levels, temperature variations, soil moisture, etc.) than the land-based functional unit.

4. Discussion

4.1. Nitrous oxide contribution to agricultural greenhouse gas emissions

In the U.S., agriculture contributes to 8.4% of the country's GHGs emissions. Although agricultural practices and local soil and weather conditions have an important effect on the GWP generated by crop production, nitrogen fertilization is generally the main source of greenhouse gases emitted from agricultural soils. Emissions of N_2O from agricultural soil management is the main source of GHG from the agricultural sector and accounts for 73.9% of the total U.S. N_2O emissions (US-EPA 2019).

The effect of cover crops on N_2O emissions is still a topic of discussion (Cavigelli et al. 2012). Research on camelina and pennycress as cover crops is still at its early stages, while rye has been extensively investigated, (Snapp et al. 2005; Blanco-Canqui et al. 2015). These authors stated that the majority of studies concluded that there is not a significant impact of rye and other non-leguminous cover crops on N_2O emissions. However, other scholars measured or estimated higher N_2O fluxes in cropping systems with rye in comparison with fallow (Petersen et al. 2011), while others observed lower N_2O fluxes (McSwiney et al. 2010). Experts agree that N_2O soil emissions are site-specific and have high spatial and temporal variability (Osborne et al. 2010; Smith 2017; Snyder et al. 2009). Additionally, available models still have high levels of uncertainty in estimating N_2O fluxes (Myrgiotis et al. 2019), and inconsistencies between laboratory simulations and field measurements have been reported (Jarecki et al. 2009). Such level of environmental variability, together with yet partial knowledge of the N-cycle dynamics, makes it difficult to clearly identify cause-effect mechanisms between cover crops and N_2O fluxes (Venterea et al. 2012).

In our study, the sequences with cover crops showed lower N-related field emissions than the control in all locations, which employed different fertilization practices. These results are in accordance with those reported by Baggs et al. (2000), Rosecrance et al. (2000), Kim et al. (2009), Snyder et al. (2009), McSwiney et al. (2010), and Basche et al. (2016). They suggested that cover crops can improve nitrogen management by curbing the release of N_2O from agricultural soils. In Ames, the sequences M/Cam-S and M/Pen-S, which received N-fertilization in spring of Year 2, had a 7% reduction of the N_2O field emissions compared with the control. The sequence M/Rye-S reduced N_2O emissions by 18% compared with the control. In Morris, M/Cam-S and M/Pen-S sequences had a 20% reduction compared with the control, while the M/Rye-S sequence had a 27% reduction compared with the control. Such estimates are in contrast with the results reported by Berti et al. (2017a) for camelina in relay-cropping with soybean. The authors observed higher N_2O field emissions in the relay system with camelina compared with soybean alone, but the study employed IPCC default emission factors. For the winter oilseed-soybean relay system, N fertilizer is typically applied in spring prior to the bolting of winter crops, to maximize oilseed yields (Gesch et al. 2014; Berti et al. 2015). Studies have demonstrated that winter oilseeds readily use the N fertilizer applied with little or no loss from the cropping system (Weyers et al., 2019; Ott et al. 2019). The results of Johnson et al. (2017) were similar to our study, who found that pennycress and camelina can significantly reduce inorganic N (NO_3-N) in the soil. However, it must be stressed that N_2O field emissions were not measured in our study. In addition, the limitations of the model employed might have overestimated the cover crop effect on N_2O fluxes in the 2-year sequence.

4.2. Nutrient losses

Proper management of N and P in agricultural soils is critical to prevent or limit eutrophication and its associated social costs (Dodds et al. 2009; Lewis et al. 2011). Agricultural soils act as non-point sources for N and P, and cultivation of maize and soybean is solely responsible for 52% of the N and 25% of the P that reach the Gulf of Mexico (Alexander et al. 2008). Kladivko et al. (2014) estimated that introducing cover crops in continuous maize and maize-soybean cropping systems in Illinois, Indiana, Iowa, Minnesota, and Ohio would potentially reduce nitrate loadings to the Mississippi River by 20%. Therefore, in this LCA study, we expected cover crops to have a positive impact on eutrophication by reducing NO_3-N runoff and leaching to surface and ground waters. The findings of the LCA study confirm this assumption. The results of the SQCB-NO_3 model showed that all three sequences with cover crops had lower nitrate leaching, but the reduction was low, ranging between 5 and 10% in Year 2. Such values are in line with the findings of Prechsl et al. (2017) and Strock et al. (2004) in southwestern Minnesota, who reported a 13% reduction of nitrate leaching by using rye as a cover crop in a maize-soybean sequence. However, a meta-analysis by Tonitto et al. (2006) found that non-legume cover crops, on average, reduce nitrate leaching by 40–70% when compared with a winter bare fallow soil. Besides the intrinsic limitations of the model employed to estimate NO_3-N leaching (Nemecek et al. 2016), the very low cover crop biomass production might have limited the NO_3-N uptake (Mohammed et al. 2020b). This last supposition seems to be confirmed by the results from Ames, where the model estimated a wider difference in nitrate leaching between the M/Rye-S and the control (both not fertilized in Year 2), and the rye aboveground biomass was the greatest measured in all experiments.

4.3. Soil loss

Water erosion is site-specific and varies due to multiple factors related to local environmental characteristics, including weather conditions, field slope, soil texture, structure and organic matter, soil cover, and field management practices (Morgan 2009). The overall results of the soil water erosion model for the three locations is in line with a USDA soil loss simulation for U.S. croplands (Potter et al. 2006). Ames and Morris sequences had higher erosion rates. The average value estimated at these locations was around 5 Mg ha^{-1} $year^{-1}$, while for the USDA simulation in the same region where the two sites are located was 5.3 Mg ha^{-1} $year^{-1}$. Prosper had much lower erosion values, an average of 0.5 Mg ha^{-1} $year^{-1}$, 1.85 Mg ha^{-1} $year^{-1}$ in the USDA simulation. The overall low soil loss in Prosper, 0.72 Mg ha^{-1} $year^{-1}$ for the control, was mainly due to the near-zero slope of the plots where the experiments were carried out.

One of the main functions of cover crops is to protect soil from erosion between the growing seasons of the main crop (Schipanski et al. 2014; Snapp et al. 2005). For all three sites (Fig. 6a), as expected, growing a winter cover crop shows a clear impact on soil erosion reduction. The living cover of camelina, pennycress, or rye in the fall, their residues left on the field during the winter, and the soil cover provided by the cover crops regrowth in spring of the following year, effectively reduced the soil losses over the 2-year period.

4.4. Residue management and SOC

The results of our study highlight the importance of correct management of the crop residues (in particular maize biomass) within the 2-year maize-soybean rotation. Given that soybean does not produce enough aboveground residue to compensate for the carbon depleted during the growing season (Adviento-Borbe et al. 2007; Johnson et al. 2006), introducing winter cover crops in a conventional system is expected to increase the SOC levels due to further addition of aboveground and belowground residues (Bonner et al. 2014; Luo et al. 2010a, 2010b). For cropping systems with maize, winter cover crops allow for higher maize residue removal rates (Kim and Dale 2005; Pratt et al. 2014; Wilhelm et al. 2010). However, in our study, the low cover crop plant density and overall biomass in Year 2 (Mohammed et al. 2020a), along with a reduction of the soybean seed yield and biomass, led to a reduction in SOC values in the M/Cam-S and M/Pen-S sequences.

According to this analysis, maize residue management is the key field operation to provide a sustainable input of residues without compromising the SOC stock over the 2-year maize-soybean rotation. Table 3 shows the aboveground maximum maize residue removal (% of total crop aboveground biomass) needed to achieve a zero-net change in SOC levels for the sequences considered in this study. Maximum residue removal to maintain SOC was 56% of the total maize residue in Ames, for the M/Rye-S sequence, while 9% for the Prosper M/Cam-S sequence was the lowest estimated removal rate, with an average value of 31% across all locations. These results are in line with Xu et al., (2019), who recommended moderate maize residue removal (<50%) to have positive SOC sequestration rates. The M/Rye-S sequence in Ames is particularly relevant since it was able to generate enough biomass to cover the SOC

Table 3
Maximum maize residue removal to maintain soil organic carbon (SOC) levels over the 2-year rotation. Maize-soybean = M-S, maize/camelina-soybean = M/Cam-S, maize/pennycress-soybean = M/Pen-S, and maize/rye-soybean = M/rye-S.

Sequence	Ames	Morris	Prosper
	% of total dry matter biomass removed		
M-S	28	33	31
M/Cam-S	30	32	9
M/Pen-S	28	29	20
M/Rye-S	56	43	33
Removal rate applied	0	70	95

debt in Year 2 and still allowed for a removal rate of >50% of maize residue. Higher maize residue removal could be achieved for the M/Cam-S and M/Rye-S sequences if the interseeded system is optimized to improve cover crop growth and biomass production.

However, particularly in areas with a short vegetative season such as the U.S. upper Midwest, the presence of abundant maize residue in the field might negatively affect cover crop establishment in the fall of Year 1 and the regrowth in the spring of Year 2 (Johnson et al. 2017). Therefore, more research is needed to identify the optimal combination of field management practices (e.g., tillage regime, planting dates, and maize residue management) to avoid a long-term SOC depletion and concurrently facilitate winter cover crop establishment.

4.5. Environmental trade-offs and economic sustainability

Having a winter cover crop, such as camelina and pennycress, that can also be harvested for oilseed, has the potential of providing an additional source of income for the farmer. If the cover crops produce enough seed yield to offset the extra cost of including a cover crop in the maize-soybean sequence and generate a higher net margin for the farmer than the conventional maize-soybean sequence, the M/Cam-S and M/Pen-S sequences could have a better performance than the control when the impact is associated with an economic functional unit ($). This was not the case for any of the locations considered in this study. This trend is clear even in the erosion category of impact (Fig. 6b), in which the effect of the cover crop generated soil loss reductions between 27% to 50% than the control when expressed per ha year^{-1} (Fig. 6a).

The main cause of this negative performance for the M/Cam-S and M/Pen-S sequences is related to a lower soybean seed yield in Year 2, compared with the control. Soybean seed yield reduction in the relay system was on average: 40% in Ames, 14.5% in Morris, and 43% in Prosper. Similar soybean yield reductions (17–42%) in soybean interseeded into camelina were reported by Gesch et al. (2014) in previous studies in Morris. Berti et al. (2015) reported 47% and 71% soybean seed yield reduction in Morris and Prosper, respectively, in relay cropping with camelina compared with sole soybean planted at the normal seeding date. However, in contrast to some studies reporting soybean yield losses offset by camelina seed yield (Gesch et al. 2014; Johnson et al. 2017, 2015; Ott et al. 2019), in our study soybean seed yield losses were not offset by the winter crops seed yield (Patel et al. 2021), causing a lower net margin. A very limited or even negligible impact of the cover crop on the main crop yield is often reported in literature, mainly because the cover crop is interseeded after the weed free critical period of the main cash crop has passed (Snapp et al. 2005). Typically, main crop yield losses are observed when the cover crop interferes with the initial vegetative phase of the main crop (Tonitto et al. 2006). The competition for resources such as nutrients, water and sunlight is arguably the reason why the seed yield of relay-cropped soybean into standing camelina or pennycress had such high reductions. Therefore, the opportunity of using relay-cropping in areas with short production windows needs to be thoroughly assessed.

The results of the present study suggest that the introduction of cover crops (such as camelina and pennycress) within the maize-soybean conventional cropping system in the U.S. upper Midwest still requires an optimization of the field management practices to ensure both environmental and economic sustainability of the cropping system (Bergtold et al. 2019; Cubins et al. 2019). More specifically, further research is needed to identify: 1) optimal seeding windows to ensure the establishment of the cover crop before the winter killing; 2) fertilization rates for the relay-cropped system that minimize nutrient losses and GHG emissions without compromising the seed yield; 3) early-maturing camelina and pennycress cultivars to reduce the time of overlapping between species in the relay-cropping system and 4) a residue management over the 2-year maize-soybean rotation to avoid a SOC depletion while not interfering with the establishment of the winter-hardy cover crop in the fall.

However, it must be stressed that the local variability both in spatial and temporal terms (e.g. site characteristics and weather conditions) can affect the results of the environmental assessment, therefore further studies are needed to confirm such findings.

5. Modelling limitations in relay-cropped systems

Life cycle assessment studies on relay-cropping systems are particularly challenging due to the complexity of modelling spatial and temporal dynamics between multiple crops in the same field. Knowledge of agronomic and ecological aspects of the interactions between species and processes that occur within an intercropped system are still not fully understood (Brooker et al. 2015). Such limitation has a direct impact on field emission models as well. To date, empirical and process-based crop models are only able to simulate a limited number of interactions within an intercropped or relay-cropped system (Gaudio et al. 2019; Tanveer et al. 2017). This often leads to modelling crops independently and not as a part of a cropping system (Oelbermann et al. 2017), therefore overlooking synergic and competitive effects between species and their combined effect on the soil biogeochemistry. This is the case when the models are employed in this study to estimate N-related field emissions. These limitations bring a high level of uncertainty into the LCA when models, and not direct measurements, are used to estimate field emissions related to biogeochemical cycles such as N_2O, NH_3, CO_2, and NO_3.

6. Conclusions

Research on winter camelina and field pennycress as cover crops in a wheat-soybean sequence has demonstrated that they have the potential to provide multiple ecosystem services and supplement farmer income when introduced in cropping systems that include soybean. However, this is the first time that relay-cropping of soybean with winter camelina and field pennycress has been studied in a maize-soybean sequence where these winter oilseeds along with the traditional cover crop, winter rye, were interseeded into maize at a late reproductive phase. Findings of this study clearly show that further research is needed to optimize the field management in a maize-soybean sequence to make these cover crops competitive with more traditional ones such as winter rye and to fully realize their ecosystem services potential.

When expressed according to the area-based functional unit, the results of the LCA showed that interseeding cover crops into a 2-year maize-soybean sequence has a clear positive impact on eutrophication potential and water erosion. Additionally, crop sequences with cover crops had lower GWP than the control (M-S) when the cover crop was not fertilized. However, the models show an overall better management of nitrogen field emissions in all locations when cover crops were used. The soil organic carbon stock was mainly affected by the maize residue management in Year 1, but cover crops did not produce a clear positive impact on SOC except for the M/Rye-S sequence, which had the least SOC losses in all locations. The sequences M/Cam-S and M/Pen-S had both less water erosion potential than the M/Rye-S sequence, likely due to their longer permanence in the field. The M/Cam-S and M/Pen-S sequences had lower eutrophication than M/Rye-S only when they were not fertilized.

The overall impact assessment results changed when expressed in the economic functional unit ($1 net margin). In the present study, where cover crops were interseeded into standing maize, crop sequences M/Cam-S and M/Pen-S were the least effective sequences to reduce GWP, eutrophication, and SOC stock compared with M/Rye-S. These results can be attributed to a lower soybean yield in the M/Cam-S and M/Pen-S sequences, which was not counterbalanced by the cover crop seed yield. This led to a lower net margin for the M/Cam-S and M/Pen-S sequences and consequently a higher environmental burden per dollar profit.

The results confirm that in an LCA study, the way the environmental performance is measured (functional unit) deeply affects the outcomes of the comparison between different treatments. For this reason, using multiple functional units allows a more comprehensive assessment of the cropping systems' performance.

Finally, some of the empirical and process-based models employed in this study have shown considerable limitations when applied to relay-cropping systems, due to the complexity of the spatial and temporal interactions between crops, which are not fully understood yet. This was particularly evident while modelling field emissions associated with N-fertilization and wind erosion. Hence, more research is needed to further study the interactions between the main crop and cover crop in relay-cropped or intercropped systems and to develop models with the capability to accurately simulate such dynamics.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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